Effect of flow rate on clogging processes in small diameter aquifer storage and recovery injection wells

by

Angela R. Thompson

Submitted to the graduate degree program in Geology
and the Graduate Faculty of the University of Kansas
in partial fulfillment of the requirements for the degree
Master of Science

_____
Dr. Randy Stotler, chair

_____
Dr. Gaisheng Liu

_____
Dr. G.L. Macpherson

_____
Dr. Jennifer A. Roberts

Date Defended: 07-14-2014

The Thesis Committee for Angela Thompson
certifies that this is the approved version of the following thesis:

Effect of flow rate on clogging processes in small diameter aquifer storage and recovery injection wells

_____
Dr. Randy Stotler, chair

Date Approved: 07-18-2014

Abstract

This study investigates injection well clogging at low injection rates (<0.38 m^3/ min or 100 gpm) through laboratory column experiments and geochemical analyses. Clogging in recharge wells is a major concern for aquifer storage and recovery (ASR) systems. Many of the current methods used to predict injection well clogging assume a higher injection rate. These methods may not adequately identify the clogging processes occurring at low flow rates.

An ongoing project at the Kansas Geological Survey (KGS) investigates a low-cost, slow flow alternative to traditional ASR recharge systems. The project utilizes gravity-induced recharge and small diameter wells installed with direct-push technology to recharge and store ground water. The KGS ASR recharge system will have significantly lower injection rates than traditional ASR wells due to increased frictional losses in small-diameter wells and the absence of injection pumps.

To examine clogging processes at low flow rates, laboratory columns packed with sand and gravel cores, taken from the Pleistocene Belleville formation during direct-push well installation at the Lower Republican River field site, were used for clogging experiments. Changes in geochemistry and hydraulic conductivity were monitored in three column sets running at flow rates of 1 m/day, 2 m/day, and 3 m/day, for seventeen days. Each set contains one column each of native aquifer water and one column of treated surface water, respectively.

Hydraulic conductivity for all columns in this experiment increased above initial levels by at least one order of magnitude. Columns with the highest pumping speed had the greatest increase in hydraulic conductivity, suggesting sufficient pressure existed to mobilize clay particles and remove them from the

columns. Experimental results and geochemical analyses indicate clay dispersion is the primary factor influencing hydraulic conductivity changes at the Republic ASR site. Observations of biofilm in effluent tubing suggest bacterial clogging of medium and low flow treated surface water columns, but bacterial abundance was not quantified in this study. Further understanding of clogging factors at low flow rates will aid in the selection of the most beneficial redevelopment and pretreatment methods for slow recharge ASR systems.

Acknowledgements

I would like to thank my mother, Mary Hyde, for putting up with me as I destroyed her beautiful garden in search of dinosaur bones and valuable minerals, for always having the best shoulder to cry on, for giving the best "this too shall pass, like a kidney stone" reassurances, and for being my own personal cheerleader. I would also like to thank my father, Robert Hyde, who taught me to always look with new eyes and who showed me what unconditional love looks like. I also thank my brother, Roger Hyde, for demonstrating that the most amazing people in this world don't need to be "normal". When I started this journey, I had these three sets of hands to hold, I wish I could hold them now as I cross the finish line.

I bestow infinite thanks upon my husband, Tommy Thompson (midnight laundry folder, "Hey kids, let's go play outside while mommy yells at the computer" speaker, partner in crime, and rock of support) for dancing with me through this ridiculous dance and for always finding a way to make me laugh when I'm ready to throw in the towel. I would like to thank my children; Julian, Annabelle, and Mackenzie Thompson; for keeping me on my toes and putting a smile on my face every morning (even at 4am).

I give many thanks to Dr. G.L. Macpherson for all the reassurances that experimental work never goes according to plan, for showing me how to do things right in the lab, and for never letting me take the easy way out (even though I complained about it at the time). I would also like to thank my advisor, Dr. Randy Stotler, for taking my sarcasm in stride and helping me navigate this wacky, wacky world of academia. I would also like to thank the Geological Society of America, Kansas Geological Foundation, University of Kansas Geology Department and Frederick T. Holden Fund for providing the essential monetary support for this project.

Many thanks are also due to the lovely Public Education ladies at the University of Kansas Natural History Museum, Teresa MacDonald and Bekkah Lampe. Thank you both for the opportunity and absolute freedom to create all those hands-on science activities I had rolling around in my head, even when subatomic physics was not my specialty. Finally I would like to thank Ferrero and Canada for manufacturing Nutella, the literal spoonful of sugar that helped the sleep deprivation go down.

Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction

As ground water levels and municipal budgets decrease, creative solutions for supplementing water resources are becoming more critical than ever. Aquifer storage and recovery (ASR) is one solution that has been implemented successfully in many areas around the world (Ziegler et al., 1999; Brown et al., 2004; Arthur et al., 2005; Kumar et al., 2006); however, the cost involved with the installation and operation of traditional large-scale ASR projects, using either surface infiltration or large-diameter well injection, may be out of reach for many small municipalities (Pyne, 1995). An ongoing project at the Kansas Geological Survey (KGS) investigates a low-cost alternative to traditional ASR recharge systems by utilizing gravity-induced recharge and a number of small-diameter wells, installed with direct-push (DP) technology, to recharge and store ground water in a near-surface Lower Republican River Valley aquifer. The proposed system has the potential to extend the availability of ASR to individual agricultural producers, small municipalities, and to aquifers with small storage capacities worldwide.

Feasibility of the new system is dependent upon the efficient transfer of surface water to the subsurface. Clogging is a major concern in many ASR systems, with the potential to severely impact system efficiency in a matter of days to weeks (Oberdorfer and Peterson, 1985; Bichara, 1988; Pyne, 1995; Baveye et al., 1998). Increased frictional losses in small-diameter wells coupled with the absence of injection pumps (gravity-induced recharge) will result in significantly lower injection rates than typical in the proposed KGS ASR system; however, few studies have examined injection well clogging at low infiltration rates. This study examines the predominant cause of clogging at low injection rates to inform future projects interested in clogging mitigation of slow-flow infiltration wells.

Ground Water Resources and ASR

With US population projected to increase by 2.4 – 2.5 million per year until 2030 and western states bearing the brunt of the current population increase in recent years (U.S. Census Bureau, 2012, 2014), innovative technologies may be required to provide water for an ever-expanding population. Though the majority of the U.S. population receives water from large municipal systems using surface water as their primary water source, ground water is the main water source for 73.5% of small community water systems in the US (U.S. Environmental Protection Agency, 2009).

If ground water supplies are depleted, small community water systems must capture and treat surface water or purchase water from a nearby community to meet water demand. Consequently, ground water rights are a contentious issue in many western states, where water demand is high and supplies are low (e.g. Schlager, 2006; Schlager and Heikkila, 2011). Furthermore, for regions designated as water scarce or water stressed, large spatial and temporal variations in water availability exist (Jury and Vaux, 2005; Oki and Kanae, 2006; Wada et al., 2010; Scanlon et al., 2012). For example, since the 1950's, the High Plains aquifer supply has decreased by 8%, but 1/3 of the decrease occurred in only 4% of the total aquifer area (Scanlon et al., 2012).

Capturing and storing water through ASR systems is a proven way to meet spatial and temporal availability challenges by buffering seasonal infiltration fluctuations, minimizing evaporative losses and providing storage in close proximity to major water users (Bloetscher et al., 2004; Jury and Vaux, 2005; Schlager, 2006). Successful ASR systems are found throughout the US, Canada, Australia, Europe, the Middle East, Asia, and Africa (e.g. Rebhun and Schwarz, 1968; Harpaz, 1971; Connorton and McIntosh, 1994; Pyne, 1995; Han, 2003; Kumar et al., 2006; Dillon et al., 2009).

While ASR systems are more cost effective than other water storage alternatives, large initial investments of time and money must be made for infrastructure. Traditional artificial recharge methods require drilling injection wells, creating pumping facilities, and constructing large recharge basins or trenches (Pyne, 1995). For example, implementation of a large-scale ASR system near Wichita, in south-central Kansas, successfully captured excess surface water to increase ground water supplies in the Equus Beds aquifer at a projected total cost of $500 million (Ziegler et al., 1999; Office of the Federal Register, 2009). For many water stressed areas in the world, the infrastructure costs for ASR systems are too high and are not offset by hydrological benefits (Jury and Vaux, 2005; Kumar et al., 2006).

Clogging and ASR

When pore spaces clog during artificial recharge, hydraulic conductivity decreases and system productivity declines. If clogging is rapid, the injection well must be redeveloped frequently, resulting in productivity declines that add to the overall cost of the ASR project. In general, wells tend to clog faster than infiltration basins, with wells clogging on a scale of days to weeks, and basins clogging on a scale of months to years (Olsthoorn, 1982; Greskowiak et al., 2005).

Clogging rates vary considerably with site geology, ground water geochemistry, infiltration method, injection water chemistry, and microbial activity (Schippers et al., 1995; Ziegler et al., 1999; Pavelic et al., 2007). Sources of injection well clogging are generally divided into three groups: physical, biological, and chemical, but groups seldom act independently in a system (Olsthoorn, 1982; Huisman and Olsthoorn, 1983; Bichara, 1986; Baveye et al., 1998; Rinck-Pfeiffer et al., 2000).

Physical clogging

The most common cause of physical clogging of recharge wells occurs when colloids and suspended solids are filtered out of injected waters by the aquifer material. Thus, physical clogging is primarily influenced by water quality, and more specifically the shape, size, composition and concentration of particles in recharge water (Bichara, 1986; Pavelic et al., 2007). Particle filtration can lead to rapid K reductions, making physical clogging the chief concern for most artificial recharge systems (Rinck-Pfeiffer et al., 2000; Rinck-Pfeiffer et al., 2002; Pavelic et al., 2011). Olsthoorn (1982) observed injection rate increases provide more particles to the system per unit time, thus speeding the clogging process. Low injection rates will supply less suspended solids to the system per unit time and may decrease the speed of physical clogging in low rate infiltration systems.

Biological clogging

Biological clogging is generally a secondary concern in artificial recharge systems (Pyne, 1995). However, biological clogging, as a general term, encompasses many different components that are very difficult to characterize and measure. Free flowing bacteria act as particulates, clogging pores in a manner similar to physical clogging by suspended particles, while the growth of attached bacterial colonies and their extracellular polymeric substances (EPS) fill pore spaces (Huisman and Olsthoorn, 1983; Baveye et al., 1998). Biofilm formation and growth is dependent upon nutrient availability, pH, temperature, mechanical and shear stresses, substrate surface area, osmotic pressure, and the presence of metabolic inhibitors (Characklis, 1981; Or et al., 2007; Pavelic et al., 2007; Gerlach and Cunningham, 2011).

Chemical clogging

Chemical clogging occurs as insoluble deposits precipitate from solution or as clays swell and disperse (Olsthoorn, 1982). Chemical clogging occurs in mixing areas as waters of differing geochemistry undergo chemical reactions during artificial recharge (Huisman and Olsthoorn, 1983). If chemical reactions near the injection well dissolve aquifer materials or disperse clays, K will increase near the well bore, but may decrease further along the flow path as minerals precipitate and clays resettle (Olsthoorn, 1982). Geochemical analysis of recharge water, ground water and aquifer material along with geochemical modeling accurately predict the potential for chemical clogging in most recharge scenarios (Pyne, 1995).

Recharge well comparison

When compared to traditionally drilled, large diameter wells, DP well installation is inexpensive and quick, with 3-4 shallow (<30m) wells installed per day. DP well installation consists of large hydraulic hammers driving small diameter (<10cm) steel rods through soils and unconsolidated materials until the desired depth is reached (typically less than 30 m). Substrate is removed from inside the rods (coring) or pushed aside (no coring), and a plastic well casing and screen is lowered into the space.

Unlike traditional wells, DP wells typically do not have a surrounding gravel pack. Clogging in both DP and traditional recharge wells occurs in the aquifer (Olsthoorn, 1982). Without a gravel pack to assist particle filtration, more suspended solids will reach the aquifer material in DP wells (Schippers et al., 1995). However, wells without a gravel pack are easier to redevelop once clogging occurs (Bichara, 1988).

Chapter 2: Materials and Methods

Study Area

The Lower Republican River (LRR) basin is located approximately two miles east of the Republic River in Republic County, Kansas (Figure 1). Core samples of the unconfined Belleville aquifer show pronounced heterogeneity in the subsurface. LRR basin lithology is generally composed of medium to coarse quartz sand, gravel, and occasional silt lenses, all of Pleistocene glaciofluvial origin (Byrne and Beck, 1950). The water table in this area exhibits a gradient of 0.95 meters per kilometer (Hansen, 1998).

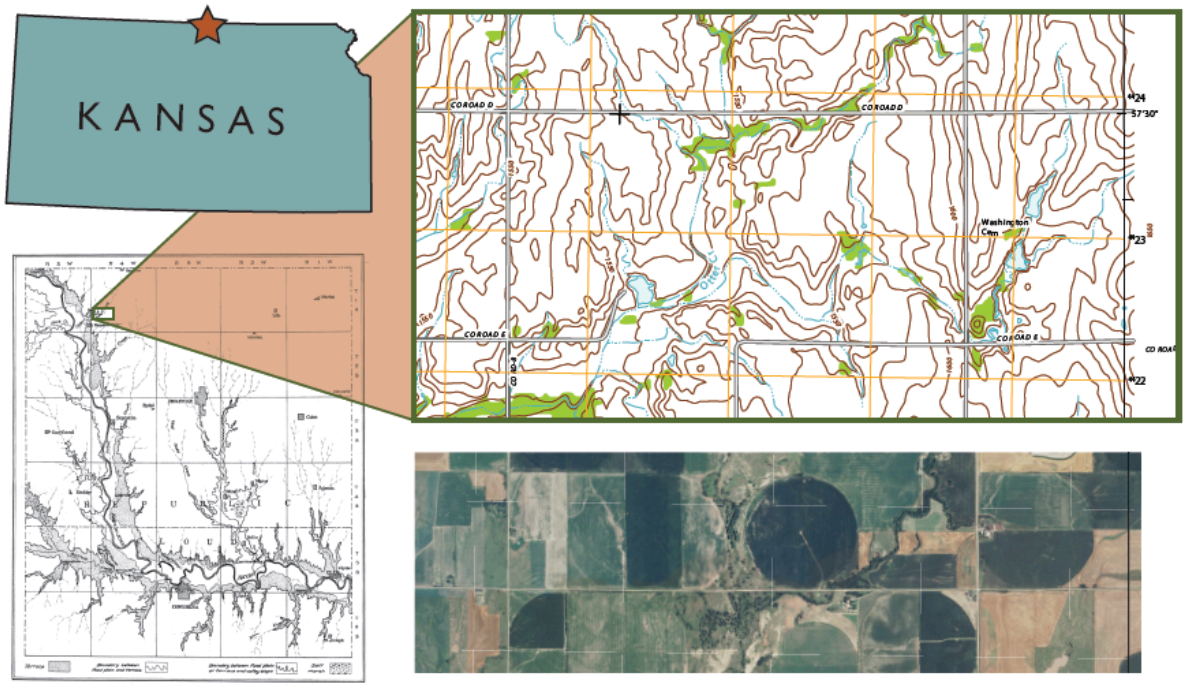

Figure 1. Map of the study area near Republic, in Republic County, Kansas with topographic map (10 ft. contour interval) and aerial photograph (Wing, 1930; U.S. Geological Survey, 2012).

Experimental Setup

This investigation (Figures 2-3) loosely follows the experimental methods of Rinck-Pfeiffer et al. (2000; 2002). Sediment cores recovered from DP well installation at the Republic site were frozen and stored prior to column assembly. Cores were cut into 13 cm sections and pushed directly into six polycarbonate columns (4.5 cm internal diameter and 13 cm length) to minimize porosity disturbances. A series of nylon mesh screens were placed under the top and bottom column end caps to support and retain the aquifer material, and diffuse influent water across the entire column area. Threaded nylon fittings wrapped with Teflon tape and stuffed with fiberglass minimized leakage at ports and retained aquifer material inside the column. Following assembly, the columns were triple wrapped in aluminum foil to eliminate photoautotrophic growth within the columns.

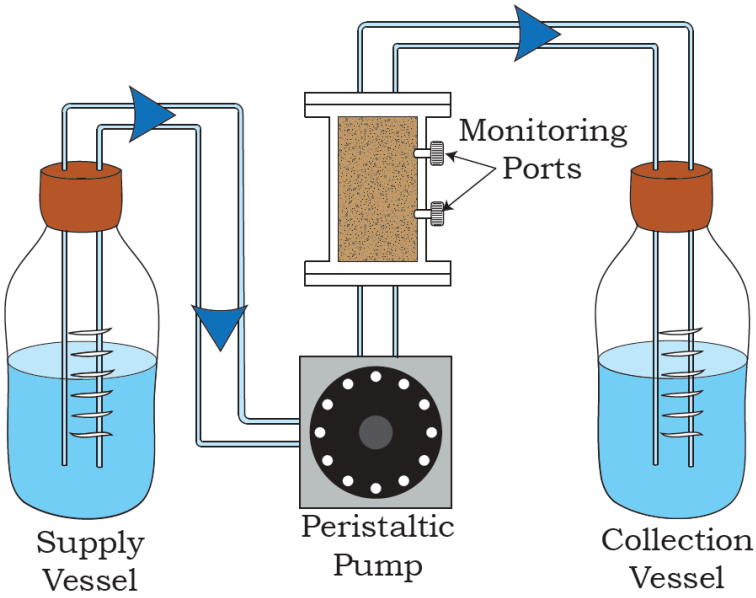

Figure 2. Diagram of experimental column set up.

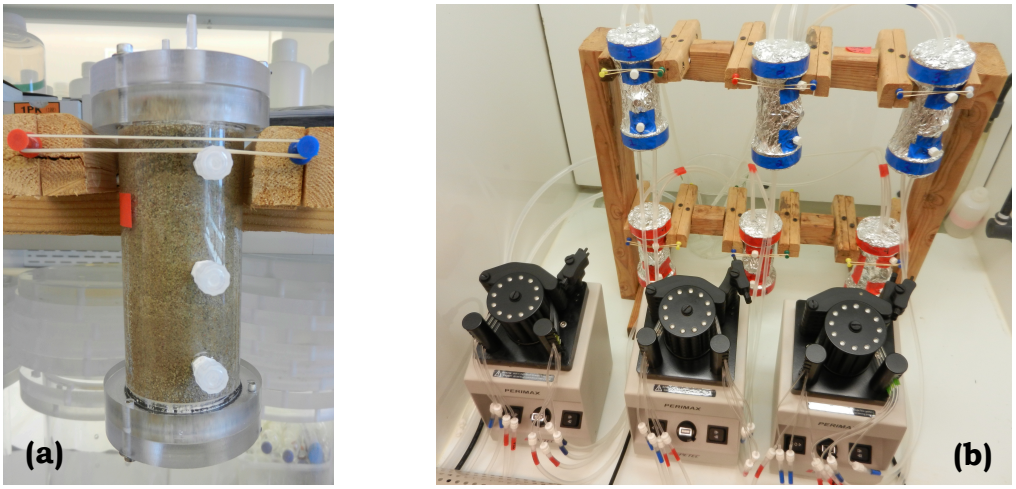

Figure 3. Photograph of column setup (a) unwrapped column filled with aquifer material from the Republic site (b) all six wrapped columns during on day 6 of the experiment.

To saturate aquifer material, a peristaltic pump slowly pushed ground water, obtained from a stock well 50 meters southeast of the installed DP well, into the columns for 48 hours. Columns were then allowed to sit, undisturbed, for 24 hours to incubate the aquifer material. After the incubation period, the peristaltic pumps were differentiated into pumping speeds of low (0.44 mL/min), medium (1.46 mL/min), and high (2.06 mL/min) rates. These pumping rates correspond to Darcy velocities of 1 m/day, 3 m/day, and 5 m/day respectively.

Three columns, one at each pumping rate, were injected with native ground water from the Republic ASR site. Three separate columns, also with one column at each pumping rate, were injected with treated surface water from the City of Lawrence Kaw River Water Treatment Plant. Thus, each column is injected with a unique combination of pumping speed and source water (summarized in Table 1).

Table 1. Experimental column groupings.

Water Type	Column #	Rate (Darcian velocity)	Clogging Process Tested
treated surface water	1	High: 4-5 m/day	physical, chemical and biological
	2	Medium: 2-3 m/day	
	3	Low >1 m/day	
native ground water	4	High: 4-5 m/day	physical and biological
	5	Medium: 2-3 m/day	
	6	Low >1 m/day	

After 12 hours of pumping, initial measurements of the hydraulic head difference between two sampling ports (Δh) and effluent discharge (Q) were measured for each column. A Fisher Scientific Traceable manometer (days 1-9) and Dwyer Series 490 wet/wet handheld digital manometer (days 10-17) were used to measure Δh in each column. Hydraulic conductivity (K) calculations were made using Darcy's Law:

$$K = \frac{Q}{A}\frac{\Delta l}{\Delta h} \qquad [1]$$

where A is the cross sectional area of the column, and Δl is the length between sampling ports.

Column flow continued for seventeen days (between 121 and 565 pore volumes, see Table 2) based on observations from other injection well clogging studies, where the most severe clogs formed between 5 and 15 days (e.g., Okubo and Matsumoto, 1979; Rinck-Pfeiffer et al., 2000; Greskowiak et al., 2005). Influent and effluent waters from each column were monitored daily for temperature, and pH. Waters were also sampled on the first day of the experimental run, and every third day thereafter for cation, anion, alkalinity, and turbidity analyses. Samples designated for alkalinity, cation and anion

analysis were filtered through a 0.45μm filter. Cation samples were also acidified to a 2% nitric acid solution.

Table 2. Column measurements and pore volume calculations.

Column area (cm^2)	Column Length (cm)	Porosity	Time (days)	Pump Speed		Pore Volumes
				Name	mL/min	
6.9	13	35%	17	Fast	2.06	565
				Medium	1.46	400
				Slow	0.44	121

Failure of the Fisher Scientific Traceable manometer early in the experiment provided unreliable K values for the first nine days. Initial K measurements were estimated using the Umwelt-Monitoring-Systeme (UMS) KSAT system. Three core samples from the Republic ASR site were packed into a KSAT sample ring, and fully saturated with tap water. The KSAT system records volumetric water flux and hydraulic head over time during a falling head permeameter test to calculate the saturated K. Each core sample was measured in triplicate and averaged to obtain initial K values.

Analyses and Data Processing

Measurements of dissolved fluoride, chloride, nitrate as nitrogen and sulfate were conducted on a Dionex 4000i ion chromatograph with detection limits of 0.01 ppm for all anions except fluoride, which has a detection limit of 0.1 ppm. A Parkin Elmer Optima 5300DV ICP-OES was used to measure dissolved calcium, magnesium, potassium, sodium, iron, silica and aluminum at KGS laboratories with detection limits of 0.01, 0.002, 0.01, 0.01, 0.005, 0.03 and 0.01 ppm respectively. Alkalinity, reported as bicarbonate, was measured by both automated titration using a Fisher Auto Titrator 390 with pH electrode and Gran titration using a Fisher Accumet AR20 pH-conductivity meter.

Determinations of turbidity were made on an Orbecco-Hellige digital direct reading turbidimeter.

The resultant data set was used for geochemical modeling in PHREEQC to calculate ionic strength and saturation indices (SI) of calcite, quartz, and iron hydroxides (Parkhurst and Appelo, 1999). Saturation indices indicate whether the water was saturated or undersaturated with respect to the minerals in question, and indicate whether mineral precipitation or dissolution could occur. Iron concentrations for both treated surface water and ground water were below ICP-OES detection limits; thus it was not possible to calculate iron hydroxide saturation indices. PHREEQC was also used to simulate the mixing of ground water and treated surface water.

Clay dispersion was estimated by calculations of sodium adsorption ratio (SAR):

$$\mathrm{SAR} = \frac{\mathrm{Na^{+}}}{\sqrt{\frac{1}{2}\left[\mathrm{Ca^{2+}} + \mathrm{Mg^{2+}}\right]}} \qquad [2]$$

and sodium percentage (SP):

$$\mathrm{SP} = \frac{\mathrm{Na^{+}}}{\mathrm{Ca^{2+}} + \mathrm{Mg^{2+}} + \mathrm{Na^{+}} + \mathrm{K^{+}}} \qquad [3]$$

where all concentrations are expressed as milliequivalents per liter. SAR values less than 6 – 4, and SP values less than 50% correlate with low levels of clay dispersion (Huisman and Olsthoorn, 1983).

Chapter 3: Results

Measured K values varied between $1.0*10^{-5}$ cm/sec and $3.6*10^{-3}$ cm/sec (Figure 4). Full data tables for this experiment are found in **Error! Reference source not found.** (Appendix I). All final K values were higher than the initial K measured by the UMS KSAT system. High injection rate columns resulted in the highest final K values. Medium and low rate treated surface water columns stabilized before day nine, and no additional K increases were observed. In contrast, K continued to increase in the ground water columns at all injection rates. The final K measurement of column #1 was one order of magnitude higher than all other treated surface water columns. Despite this, column #1 was the only one where K was observed to decline after an initial K increase.

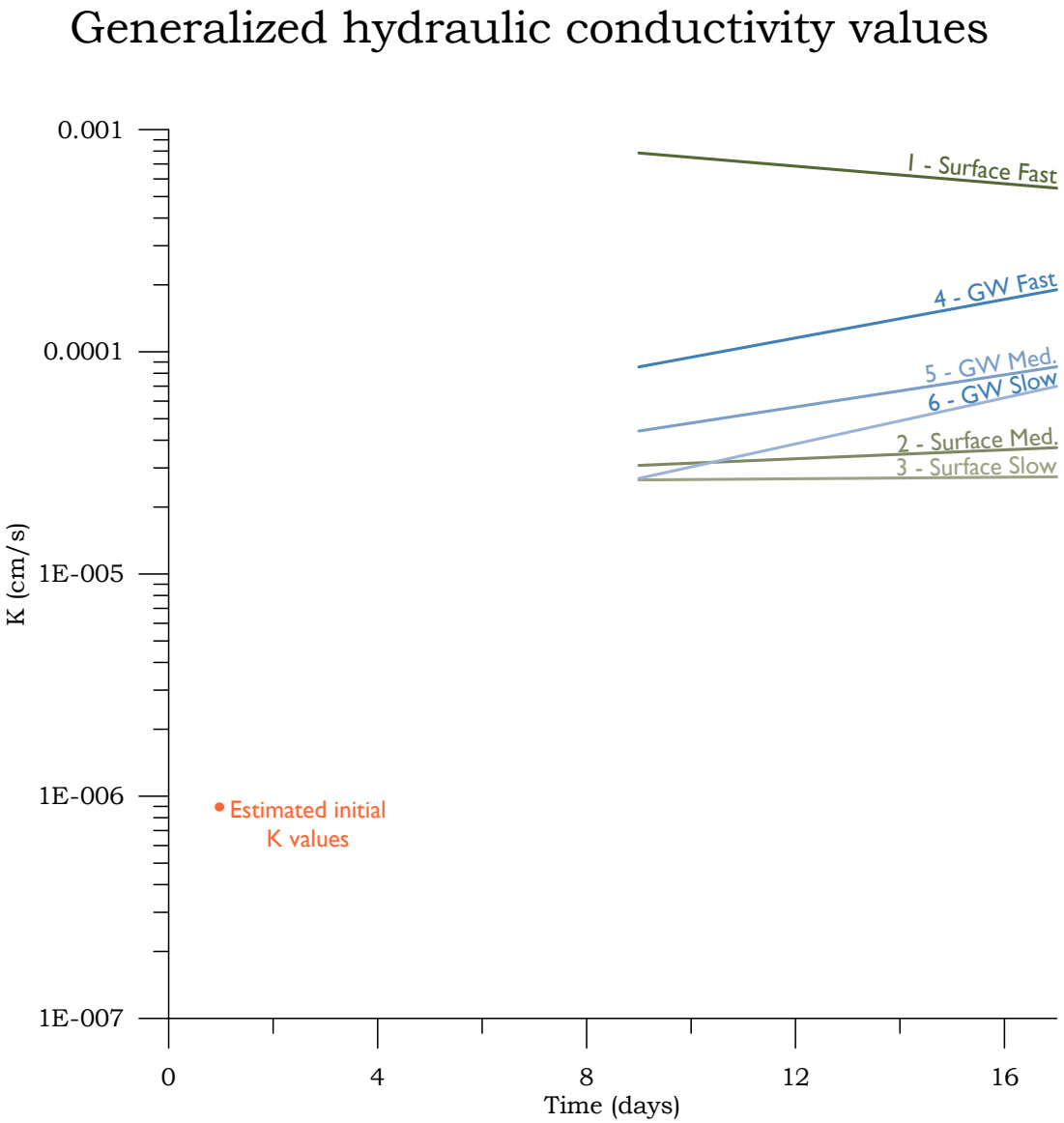

Figure 4. Generalized hydraulic conductivity results from all three experimental columns, see Table 1. Points at day nine and day seventeen are calculated by averaging the three nearest K values. Initial K values for all columns were measured with repacked core in the UMS KSAT system.

Specific conductance, mineral saturation indices, and SAR and SP values varied between the treated surface water and ground water (Table 3); however these measures did not vary significantly between influent and effluent water samples for each column. The treated surface water is a moderately conductive (972.1 μS/cm) sodium chloride water slightly undersaturated with respect to both calcite (SI = -0.40) and quartz (SI = -0.24) (Table 3). Calculated SAR and SP values for influent surface water are 3.84 and 61.4% respectively. The ground water is a less conductive (222.9 μS/cm) calcium bicarbonate water near equilibrium (slightly oversaturated) with respect to quartz (SI = 0.43), but undersaturated with respect to calcite (SI = -1.40). Calculated SAR and SP values for influent ground water are 0.32 and 16.6% respectively. The modeled 1:1 mixture of surface and ground water is an intermediately conducitve (588.4 μS/cm) sodium bicarbonate water undersaturated with respect to calcite (SI = -0.92), and slightly oversaturated with respect to quartz (SI = 0.22). Calculated SAR and SP values for mixed water are 2.39 and 53.8 respectively.

All columns produced cloudy effluent throughout the experiment, with the effluent from column #6 exhibiting the most pronounced cloudiness. Pink biofilm observed in effluent and influent tubing of columns 2, 3 and 6, indicate biological metabolism. Pink biofilm completely clogged effluent tubes of column #3 on day 10 and 14. Replacement of effluent tubes was necessary to reestablish water flow for discharge measurements. Pink biofilm was observed only in the influent and effluent tubing and was not visible in the column material itself.

Frequent episodes of leaking, peristaltic pump failure and tubing malfunctions produced spikes and dips in K measurements (Figure 7). Column leaks became so severe in column #7 on day 11 that removal and replacement of the end cap at the bottom of the column became necessary.

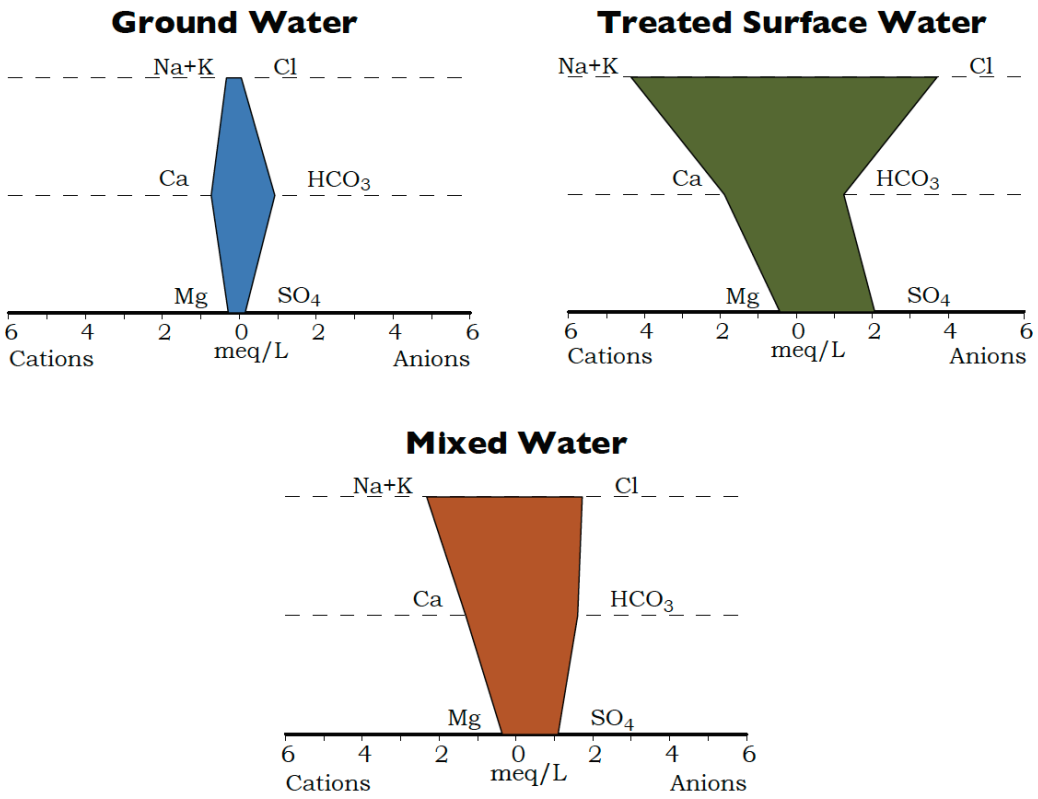

Figure 5. Stiff diagrams showing the composition of ground water, treated surface water and a simulated equal mixture of both waters.

Table 3. Water properties of initial treated surface water, initial ground water and PHREEQC simulated 1:1 mixture of both waters.

	Treated Surface Water	Mixed Water (modeled)	Ground Water
Calcite SI	-0.40	-0.92	-1.40
Quartz SI	-0.24	0.22	0.43
pH	7.70	7.34	7.10
Conductivity (µS/cm)	959.2	588.4	222.9
TDS (ppm)	479.6	294.2	111.5
Temp (°C)	20.0	20.1	20.1
HCO3- (ppm)	76	67	57
Al (ppm)	0.01	0.01	0.01
Ca (ppm)	38	26	15
Cl (ppm)	150	78	1.6
F (ppm)	0.9	0.5	0.1
K (ppm)	10	8	5
Mg (ppm)	5.3	4.4	3.6
NO3-N (ppm)	0.40	1.4	2.5
Na (ppm)	95	50	5.3
SO4 (ppm)	100	54	7.1
Si (ppm)	3.1	4.1	14
SAR	3.8	2.4	0.32
SP (%)	61	54	17

Chapter 4: Discussion

Contrary to expectations, an increase in K was observed in all six columns. However, final K values show a clear differentiation between both influent water type and injection speed. Final K values for high rate columns are all higher than low rate columns, and the two lowest rate treated surface water columns have the lowest final K values of the experiment. Due to variations in packing technique, the initial K measurement, conducted with the UMS KSAT system, may provide a lower value than the true initial K. The UMS KSAT system uses a column much larger in diameter than the collected DP core. To load the UMS KSAT column, sediment had to be removed and repacked prior to testing. This could account for some of the observed increase in K.

Different clogging processes affected ground and surface water columns. Similar rates of K increase for ground water columns at all pumping speeds suggest the same processes are acting on all three columns. These columns represent the influence of physical and, to a lesser degree, biological, change over time, since geochemistry remained unchanged. In contrast, treated surface water columns were influenced by all three potential clogging sources (physical, chemical and biological). Steady K values, SP calculations and observations of biological activity in effluent tubing suggest clay swelling and biological clogging in columns #2 and #3 (surface water, medium and low flow rates) are sufficient to offset the K increases observed in columns #4-6 (ground water, high, medium, and low flow rates).

As mentioned previously, columns #1 and #4, pumping at the highest rate, displayed the largest K increase for each water type. Thus, physical movement of suspended solids and rearrangement of aquifer material due to pumping pressure are likely responsible for increasing K in ground water columns. Aquifer clays and suspended solids in these columns were subjected to the greatest pressure and were likely removed from the column at a higher rate. While K increased due to the removal of aquifer material in 13 cm length columns, there is no guarantee the entrained particles will stay suspended. More particles suspended initially mean more particles are available to settle out further in the aquifer, leading to a decrease in K down the flow path.

Geochemical analyses of water samples indicate treated surface water is more mineralized with respect to ground water (Figure 5). Both treated surface water and ground water are undersaturated with respect to calcite, suggesting K reductions due to calcite precipitation are not a concern at this site. If calcite is present in the aquifer, surface water infiltration could increase hydraulic conductivity by dissolving calcite. Although ground water is slightly oversaturated with respect to quartz, when ground water is mixed with treated surface water, the mixture is closer to equilibrium with respect to quartz. Stable concentrations of major ions in the influent and effluent waters throughout the experiment suggest no large precipitation or dissolution occurred within the columns.

Treated surface water SAR and SP calculations indicate a geochemical potential for clay dispersion. While SAR levels are below 4, they fall within the upper limit of acceptable SAR values. SP values are well above 50%, indicating a high potential for clay dispersion (Figure 6). PHREEQC simulation of groundwater mixing shows a rapid SP increase in the initial mixing stages. With only a mixture of 20% surface water in ground water, SP increases from 16.6% (in ground water) to 41.2% (in the mixed water). Since surface water has a SP value of 61.4%, this SP increase with a small mixture of surface water accounts

for 55% of the total possible SP increase as surface water mixes with ground water (see Table 5 in Appendix I).

Biological activity was observed in the two low speed columns and the medium speed treated surface water column (#2, #3 and #6). Since the K increases were smallest in these three columns, biological activity at lower flow rates likely negatively affected K. Although biofilm was observed during this experiment, quantification of the biological influence on clogging was not conducted in this study and should be investigated in further studies. Luckily, of the three types of clogging to affect recharge wells, biological is by far the easiest to remediate (Huisman and Olsthoorn, 1983; Pyne, 1995).

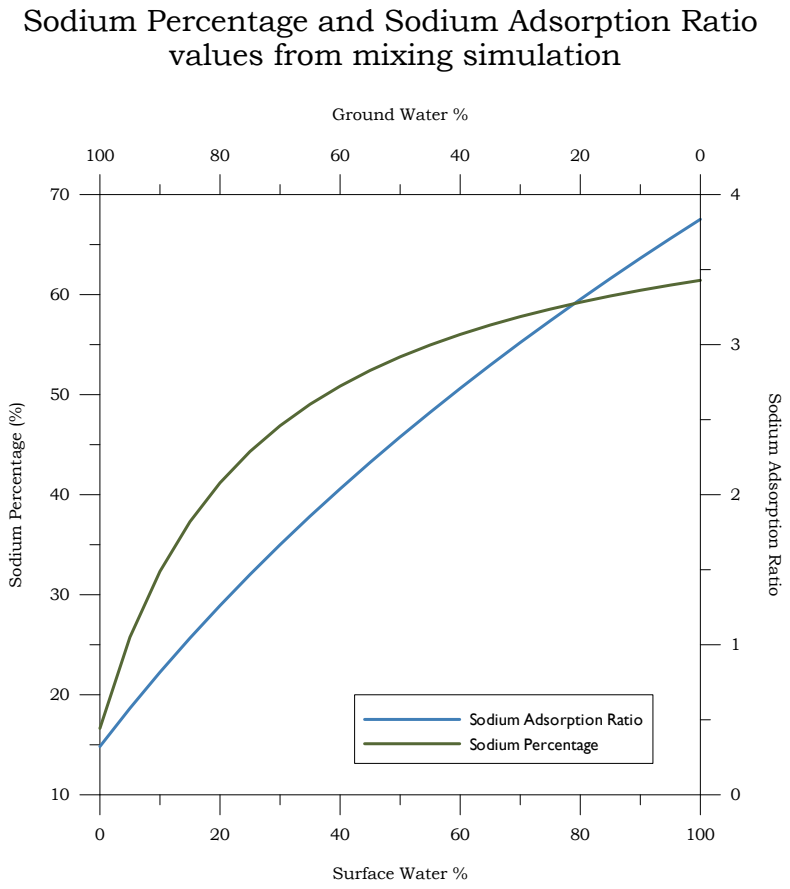

Figure 6. SAR and SP values from PHREEQC simulated mixing of surface water with ground water.

Cloudy effluent and high SP values in column experiments suggest dispersion and mobilization of clay particles is possible at the Republic ASR study area. While clay dispersion increased K in the 13 cm length columns, dispersed clay particles in the field may settle, accumulate and reduce K further in the aquifer than can be simulated with this experiment. Future field experiments at the Republic site should be aware that clogging through the swelling and dispersing clays may offset any reductions in physical clogging due to low infiltration rates. Thus, clogging prediction methods that examine only physical or biological sources and neglect chemical analysis of potential clay dispersion may provide an inaccurate assessment of clogging potential.

The most common methods to predict clogging in recharge wells are membrane filtration index (MFI) tests, and assimilable organic carbon (AOC) measurements (Schippers et al., 1995; Dillon et al., 2001; Bouwer, 2002), and numerical and analytical models (Vandevivere et al., 1995; Baveye et al., 1998; Seki and Miyazaki, 2001; Ebigbo et al., 2012) do not account for clay dispersion. MFI tests examine K reduction when physical clogging is the primary source of clog formation. The method, proposed by Schippers and Verdouw (1980), uses a 0.45μm filter and high flow rate to examine the clogging potential of a particular water sample. To measure AOC, bacteria are plated, incubated with the water sample, and growth is measured. Results are then compared to growth in acetate solutions of known carbon content (Bouwer, 2002). Finally, several numerical models exist to predict recharge well clogging; however, many of these models focus on a single process (e.g., Vandevivere et al., 1995; Seki and Miyazaki, 2001; Ebigbo et al., 2012) and require extensive empirical parameters that are difficult to measure (Baveye et al., 1998).

Few studies exist with which to compare the results of this study. Rinck-Pfeiffer et al. (2000; 2002) observed a K increase on the tenth day of similar column experiments; however, these experiments utilized a sandy limestone core material permeated with reclaimed waste water. In this instance, the

authors attributed K increases to calcite dissolution. There is no evidence to suggest K increases in the present experiment are a result of mineral dissolution. Hutchinson (1993) investigated traditional high flow injection well clogging at a field site with similar geology to the Republic ASR site, but did not observe K increases during the experiment.

Existing injection well clogging experiments that examine multiple clogging mechanisms usually investigate clogging by injection of reclaimed waste water in limestone aquifers (e.g. Rinck-Pfeiffer et al., 2002). Results of experiments using low flow injection in small-diameter wells at these sites will likely differ significantly from treated river water in an unconsolidated quartz sand and clay aquifer. Care must be taken when applying the results of this study to other slow-flow ASR sites. The predominant process affecting K at the Republic ASR site is a factor of the site's unique lithology, ground water chemistry, injection water chemistry and flow rate. Thus, future slow-flow ASR projects need to consider the unique characteristics of the site to determine the dominant clogging process.

Chapter 5: Conclusions

Experimental results suggest hydraulic conductivity values are affected by injection rate. Further experimentation is necessary to understand all of the variables influencing observed K changes. In this experiment, K changes are dominated by mobilization of clay particles through geochemical and physical processes.

Since all existing methods and models used to predict injection well clogging focus on a single clogging mechanism, selection of the best method to predict injection well clogging depends on the system's primary clogging source: physical, biological or chemical. While physical factors are assumed as the primary clogging mechanism in many injection wells, this experiment

highlights a situation where chemical and bacterial clogging are more significant. This study demonstrates the need for a greater understanding of the predominant pore-scale interactions at low injection rates for different project sites.

Chapter 6: Recommendations and Future Study

Further study of the effect of physical, biological and chemical processes on clog formation would benefit further low injection rate ASR projects. Future studies would greatly benefit from improved laboratory equipment for experimental success. Of greatest importance are reliable peristaltic pumps, tubes connections and fittings able to withstand moderate pressures, and water-tight columns. Pressure transducers inserted into column monitoring ports would minimize pressure disturbances, and provide more reliable data.

Future studies should include mineralogical investigations via X-ray diffraction of aquifer material and measurement of total suspended solids (TSS) to identify swelling clays and soluble aquifer material. TSS should be measured via membrane filtration methods, such as MFI, to determine TSS of influent and effluent waters. TSS measurements will also allow correlations between TSS concentration and K reduction over time. It is worth noting, MFI is not enough to predict clogging potential when biological clogging is prominent (Schippers et al., 1995). Also, modifications on the MFI method use high flow rates comparable to expected pumping rates in traditional injection wells, or higher, to simulate physical clogging processes (Hutchinson, 1993; Schippers et al., 1995; Dillon et al., 2001). Methods utilizing high flow rates to characterize clogging rates may not accurately represent clogging mechanisms at low infiltration rates.

Air entrainment (also considered a physical clogging mechanism) occurs when free falling water entraps gas bubbles or when water pressure drops below

atmospheric pressure and degassing occurs (Olsthoorn, 1982). As the gas bubbles are pushed into the aquifer, they clog pore throats and reduce K. However, air entrainment is of greatest concern when injection rates and injection pressures are high, and should not be a concern for future slow-flow experiments.

The contribution of bacterial colonies and their surrounding EPS, together termed biofilm, to K reduction is particularly difficult to quantify. Some measures of biomass quantify bacterial colonies through metabolic activity, but do not take into account dead microbes and EPS (Gerlach and Cunningham, 2011). Since EPS are not metabolically active, and contribute a large portion of overall biofilm volume for some bacteria, procedures that focus on metabolic measurements may underestimate bacterial clogging potential (Vandevivere and Baveye, 1992; Wu et al., 1997; Pinkart et al., 2002; Pavelic et al., 2011). Additionally, bacterial abundances may change with depth below the water table (Roudnew et al., 2012). The combination of all these factors make assessments of bacterial clogging at sites with differing ground water compositions, lithology, and recharge water compositions extremely difficult. In several studies, fluid flow rate is tied to bacterial growth in saturated porous media. Characklis (1981) and Okubo and Matsumoto (1979) both show lower infiltration rates result in more complete bacterial nutrient removal. Rinck-Pfeiffer et al. (2002) observed bacterial colonies located deeper within experimental columns and more pronounced biological clogging at low flow rates. Furthermore, limits on bacterial growth by fluid shear stresses are minimized by low flow rates (Taylor and Jaffé, 1990; Pavelic et al., 2011).

To assess bacterial influences on injection well clogging, future studies should monitor chlorine concentration, as well as AOC and dissolved oxygen levels for both influent and effluent waters to investigate the influence of biological activity. AOC measurements indicate the potential for bacterial growth given no shear stresses acting on the system and unlimited time to process

nutrients. The aqueous environment formed at low injection rates more accurately represents the environment in which AOC is measured

Field study of injection well clogging would allow comparison with experimental results. Future field investigations should monitor changes in specific capacity and injection well pressure over time; however, subsurface heterogeneities at the field site may make clogging rate comparisons difficult.

References

Arthur, J., Dabous, A., and Cowart, J., 2005, Water–rock geochemical considerations for aquifer storage and recovery: Florida case studies: Developments in Water Science, v. 52, p. 327-339.

Baveye, P., Vandevivere, P., Hoyle, B. L., DeLeo, P. C., and de Lozada, D. S., 1998, Environmental impact and mechanisms of the biological clogging of saturated soils and aquifer materials: Critical Reviews in Environmental Science and Technology, v. 28, no. 2, p. 123-191.

Bichara, A., 1986, Clogging of Recharge Wells by Suspended Solids: Journal of Irrigation and Drainage Engineering, v. 112, no. 3, p. 210-224.

Bichara, A., 1988, Redevelopment of clogged recharge wells: Journal of Irrigation and Drainage Engineering, v. 114, no. 2, p. 343-350.

Bloetscher, F., Muniz, A., and Witt, G. M., 2004, Groundwater injection: modeling, risks, and regulations, McGraw-Hill Professional.

Bouwer, H., 2002, Artificial recharge of groundwater: hydrogeology and engineering: Hydrogeology Journal, v. 10, no. 1, p. 121-142.

Brown, C., Hatfield, K., and Newman, M., Lessons learned from a review of 50 ASR projects from the United States, England, Australia and India, *in* Proceedings UCOWR/NIWR Annual Conference: Increasing Freshwater Supplies. Santa Fe, NM2004.

Byrne, F. E., and Beck, H. V., 1950, Geologic Construction-material Resources in Republic County, Kansas, US Department of the Interior, Geological Survey.

Characklis, W., 1981, Bioengineering report: Fouling biofilm development: A process analysis: Biotechnology and Bioengineering, v. 23, no. 9, p. 1923-1960.

Connorton, B. J., and McIntosh, P., Artificial Groundwater Recharge II : EUREAU Survey on Aritificial recharge, *in* Proceedings Proceedings of the second international symposium on artificial recharge of ground water: Walt Disney World, Swan, FL July 17-221994.

Dillon, P., Pavelic, P., Massmann, G., Barry, K., and Correll, R., 2001, Enhancement of the membrane filtration index (MFI) method for determining the clogging potential of turbid urban stormwater and reclaimed water used for aquifer storage and recovery: Desalination, v. 140, no. 2, p. 153-165.

Dillon, P., Pavelic, P., Page, D., Beringen, H., and Ward, J., 2009, Managed aquifer recharge: an introduction, National Water Commission Canberra, Australia.

Ebigbo, A., Phillips, A., Gerlach, R., Helmig, R., Cunningham, A. B., Class, H., and Spangler, L. H., 2012, Darcy-scale modeling of microbially induced carbonate mineral precipitation in sand columns: Water Resources Research, v. 48, no. 7.

Gerlach, R., and Cunningham, A. R., 2011, Influence of Biofilms on Porous Media Hydrodynamics, *in* Vafai, K., ed., Porous Media: Applications in Biological Systems and Biotechnology: Boca Raton, CRC Press, p. 173-230.

Greskowiak, J., Prommer, H., Massmann, G., Johnston, C. D., Nützmann, G., and Pekdeger, A., 2005, The impact of variably saturated conditions on hydrogeochemical changes during artificial recharge of groundwater: Applied Geochemistry, v. 20, no. 7, p. 1409-1426.

Han, Z., 2003, Groundwater resources protection and aquifer recovery in China: Environmental Geology, v. 44, no. 1, p. 106-111.

Hansen, C. V., 1998, Water-table conditions, aquifer properties, and streambed permeability along the Republican River from near Hardy, Nebraska, to Concordia, Kansas: Lawrence, KS, US Department of the Interior, US Geological Survey.

Harpaz, Y., 1971, Artificial ground-water recharge by means of wells in Israel: Journal of the Hydraulics Division, v. 97, no. 12, p. 1947-1964.

Huisman, L., and Olsthoorn, T. N., 1983, Artificial Groundwater Recharge, Boston, Massachusetts, Pitman Publishing, 320 p.:

Hutchinson, A. S., 1993, Estimation and quantification of injection well clogging, Tucson, Arizona.

Jury, W. A., and Vaux, H., 2005, The role of science in solving the world's emerging water problems: Proceedings of the national academy of sciences of the united states of america, v. 102, no. 44, p. 15715-15720.

Kumar, M. D., Ghosh, S., Patel, A., Singh, O. P., and Ravindranath, R., 2006, Rainwater harvesting in India: some critical issues for basin planning and research: Land Use and Water Resources Research, v. 6, no. 1, p. 1-17.

Oberdorfer, J. A., and Peterson, F. L., 1985, Waste-Water Injection: Geochemical and Biogeochemical Clogging Processes: Ground Water, v. 23, no. 6, p. 753-761.

Office of the Federal Register, N. A. a. R. A., 2009, Federal register Volume 74, Issue 229, *in* Department of the Interior, B. o. R., ed., p. 62807 - 62808.

Oki, T., and Kanae, S., 2006, Global hydrological cycles and world water resources: science, v. 313, no. 5790, p. 1068-1072.

Okubo, T., and Matsumoto, J., 1979, Effect of infiltration rate on biological clogging and water quality changes during artificial recharge: Water Resources Research, v. 15, no. 6, p. 1536-1542.

Olsthoorn, T. N., 1982, Clogging of recharge wells: main subjects: KIWA-communications, v. 72.

Or, D., Smets, B. F., Wraith, J., Dechesne, A., and Friedman, S., 2007, Physical constraints affecting bacterial habitats and activity in unsaturated porous media–a review: Advances in Water Resources, v. 30, no. 6, p. 1505-1527.

Parkhurst, D. L., and Appelo, C., 1999, User's guide to PHREEQC (Version 2): A computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations.

Pavelic, P., Dillon, P., Mucha, M., Nakai, T., Barry, K., and Bestland, E., 2011, Laboratory assessment of factors affecting soil clogging of soil aquifer treatment systems: Water research, v. 45, no. 10, p. 3153-3163.

Pavelic, P., Dillon, P. J., Barry, K. E., Vanderzalm, J. L., Correll, R. L., and Rinck-Pfeiffer, S. M., 2007, Water quality effects on clogging rates during reclaimed water ASR in a carbonate aquifer: Journal of Hydrology, v. 334, no. 1, p. 1-16.

Pinkart, H., Ringelberg, D., Piceno, Y., Macnaughton, S., and White, D., 2002, Biochemical approaches to biomass measurements and community structure analysis: Manual of environmental microbiology, v. 2, p. 101-113.

Pyne, R. D. G., 1995, Groundwater Recharge and Wells: A Guide to Aquifer Storage Recovery, Lewis Publishers, 376 p.:

Rebhun, M., and Schwarz, J., 1968, Clogging and contamination processes in recharge wells: Water Resources Research, v. 4, no. 6, p. 1207-1217.

Rinck-Pfeiffer, S., Dillon, P., Ragusa, S., and Hutson, J., 2002, Injection well clogging processes during aquifer storage and recovery (ASR) with reclaimed water: Management of Aquifer Recharge for Sustainability; proceedings of the 4th International Symposium on Artificial Recharge of Groundwater, v. 4, p. 189-194.

Rinck-Pfeiffer, S., Ragusa, S., Sztajnbok, P., and Vandevelde, T., 2000, Interrelationships between biological, chemical, and physical processes as an analog to clogging in aquifer storage and recovery (ASR) wells: Water Resources, v. 34, no. 7, p. 2110-2118.

Roudnew, B., Seymour, J. R., Jeffries, T. C., Lavery, T. J., Smith, R. J., and Mitchell, J. G., 2012, Bacterial and Virus-Like Particle Abundances in Purged and Unpurged Groundwater Depth Profiles: Ground Water Monit. Remediat., v. 32, no. 4, p. 72-77.

Scanlon, B. R., Faunt, C. C., Longuevergne, L., Reedy, R. C., Alley, W. M., McGuire, V. L., and McMahon, P. B., 2012, Groundwater depletion and sustainability of irrigation in the US High Plains and Central Valley: Proceedings of the national academy of sciences, v. 109, no. 24, p. 9320-9325.

Schippers, J., and Verdouw, J., 1980, The modified fouling index, a method of determining the fouling characteristics of water: Desalination, v. 32, p. 137-148.

Schippers, J., Verdouw, J., and Zweere, G., 1995, Predicting the clogging rate of artificial recharge wells: Journal of Water Supply Research and Technology-Aqua, v. 44, no. 1, p. 18-28.

Schlager, E., 2006, Challenges of governing groundwater in US western states: Hydrogeology journal, v. 14, no. 3, p. 350-360.

Schlager, E., and Heikkila, T., 2011, Left High and Dry? Climate Change, Common-Pool Resource Theory, and the Adaptability of Western Water Compacts: Public Administration Review, v. 71, no. 3, p. 461-470.

Seki, K., and Miyazaki, T., 2001, A mathematical model for biological clogging of uniform porous media: Water Resources Research, v. 37, no. 12, p. 2995-2999.

Taylor, S. W., and Jaffé, P. R., 1990, Biofilm growth and the related changes in the physical properties of a porous medium: 1. Experimental Investigation: Water resources research, v. 26, no. 9, p. 2153-2159.

U.S. Census Bureau, P. D., 2012, 2012 National Projections.

-, 2014, Estimates of the Components of Resident Population Change: April 1, 2010 to July 2013.

U.S. Environmental Protection Agency, O. o. W., 2009, 2006 Community Water System Survey, v. I.

U.S. Geological Survey, N. G. T. O. C., 2012, USGS 7.5-minute image map for Republic Kansas: United States Department of the Interior.

Vandevivere, P., and Baveye, P., 1992, Effect of bacterial extracellular polymers on the saturated hydraulic conductivity of sand columns: Applied and Environmental Microbiology, v. 58, no. 5, p. 1690-1698.

Vandevivere, P., Baveye, P., Lozada, D. S., and DeLeo, P., 1995, Microbial clogging of saturated soils and aquifer materials: Evaluation of mathematical models: Water Resources Research, v. 31, no. 9, p. 2173-2180.

Wada, Y., van Beek, L. P., van Kempen, C. M., Reckman, J. W., Vasak, S., and Bierkens, M. F., 2010, Global depletion of groundwater resources: Geophysical Research Letters, v. 37, no. 20.

Wing, M. E., 1930, The Geology of Cloud and Republic Counties, Kansas, *in* Survey, K. G., ed., Volume Bulletin 15.

Wu, J., Gui, S., Stahl, P., and Zhang, R., 1997, Experimental study on the reduction of soil hydraulic conductivity by enhanced biomass growth: Soil science, v. 162, no. 10, p. 741-748.

Ziegler, A. C., Christensen, V. G., and Ross, H. C., 1999, Baseline Water Quality and Preliminary Effects of Artificial Recharge on Ground Water, South Central Kansas, 1995-98, US Depertment of the Interior, US Geological Survey.

Appendix I. Column Experiment Data

Table 4. Complete experimental results for days 3 through 17

	Day	K (cm/sec)	pH	Temp (°C)	SpC† (µS/cm)	TDS† (ppm)	HCO_3^- (ppm)	Al (ppm)	Ca (ppm)	Cl (ppm)	F (ppm)	Fe (ppm)	K (ppm)	Mg (ppm)	NO_3-N (ppm)	Na (ppm)	SO_4 (ppm)	Si (ppm)
Column 1 - Treated Surface Water, Fast	3		7.65	23.1														
	4		7.80	21.9	735.6	473.5	74	0.02	36	150	0.8	<0.005	10	6.0	0.40	93	100	3.5
	5		7.40	22.7														
	6		7.28	22.5														
	7		7.71	22.0														
	8		7.40	22.0														
	9	9.80E-05	7.57	22.5	726.0	444.8	74	0.02	38	120	0.8	<0.005	11	5.2	0.60	94	98	3.8
	10	3.62E-03	7.06	21.6														
	11	5.49E-04	7.76	22.3														
	12	6.86E-04	7.16	21.5														
	13	3.71E-04	7.34	22.2														
	14	2.86E-04	7.29	21.4														
	15	5.68E-04	7.05	21.3														
	16	6.10E-04	7.45	21.8														
	17	4.58E-04	7.47	21.9	726.9	446.8	77	0.01	38	120	0.7	<0.005	10	5.2	0.40	93	99	3.3
	Average		7.43	22.0	729.5	455.0	75	0.02	37	130	0.8	<0.005	10	5.5	0.50	94	99	3.5
Column 2 - Treated Surface Water, Med.	3																	
	4		7.88	21.7	668.3	441.0	70	0.02	33	140	0.7	<0.005	9.6	7.3	0.40	82	93	4.9
	5		7.76	22.0														
	6		7.32	21.7														
	7		7.25	21.2														
	8		7.20	21.8														
	9	3.52E-05	7.28	22.3	728.9	450.2	79	0.02	38	120	0.7	<0.005	10	5.3	0.70	94	98	3.9
	10	4.06E-05	7.28	21.4														
	11	3.79E-05	7.48	21.7														
	12	3.91E-05	7.33	21.3														
	13	4.05E-05	7.34	22.1														
	14	3.60E-05	7.81	21.8														
	15	3.85E-05	7.39	21.6														
	16	3.54E-05	7.33	21.6														
	17	3.72E-05	7.45	22.5	725.8	444.0	75	0.01	38	120	0.7	<0.005	10	5.2	0.40	93	98	3.3
	Average		7.44	21.8	707.7	445.1	75	0.01	36	130	0.7	<0.005	10	5.9	0.50	90	96	4.0

† Total dissolved solids were calculated by adding all measured ions, not from specific conductance measurements.

	Day	K (cm/sec)	pH	Temp (°C)	SpC† (µS/cm)	TDS† (ppm)	HCO_3^- (ppm)	Al (ppm)	Ca (ppm)	Cl (ppm)	F (ppm)	Fe (ppm)	K (ppm)	Mg (ppm)	NO_3-N (ppm)	Na (ppm)	SO_4 (ppm)	Si (ppm)
Column 3 - Treated Surface Water, Slow	3																	
	4		7.51	22.3	640.1	418.4	71	0.01	34	130	0.7	<0.005	9.1	6.3	0.30	76	85	5.8
	5		7.50	22.1														
	6		7.37	22.0														
	7		7.34	22.2														
	8		7.47	22.2														
	9	2.93E-05	7.39	22.7	728.5	442.8	74	0.01	38	120	0.7	<0.005	10	5.4	0.70	94	98	1.9
	10	1.89E-05	7.41	21.9														
	11	1.05E-05	7.64	21.7														
	12	2.29E-05	7.49	21.8														
	13	1.37E-05	7.70	22.0														
	14	1.88E-05	7.43	21.8														
	15	1.92E-05	7.46	21.5														
	16	3.05E-05	7.46	21.7														
	17	3.23E-05	7.50	22.5	721.6	445.4	76	0.01	38	120	0.7	<0.005	10	5.5	0.40	94	97	3.5
	Average		7.48	22.0	696.7	435.5	74	0.01	36	120	0.7	<0.005	9.9	5.7	0.50	88	93	4.6
Treated Surface Water - Source	3		7.52	20.4														
	4		7.70	20.0	709.4	479.6	76	0.01	38	150	0.9	<0.005	10	5.3	0.40	95	100	3.1
	5		7.71	20.4														
	6		7.80	20.4														
	7		7.64	20.0														
	8		7.69	20.5														
	9		7.71	20.5	726.6	448.3	79	0.03	38	120	0.7	<0.005	10	5.2	0.50	93	98	3.5
	10		7.73	20.2														
	11		7.73	20.1														
	12		7.60	20.0														
	13		7.57	19.8														
	14		7.62	20.0														
	15		7.55	20.1														
	16		7.61	20.0														
	17		7.25	20.1	721.6	443.1	74	0.12	38	120	0.7	<0.005	10	5.2	0.50	93	98	3.2
	Average		7.63	20.2	719.2	457.0	76	0.05	38	130	0.8	<0.005	10	5.2	0.50	94	99	3.3

	Day	K (cm/sec)	pH	Temp (°C)	SpC† (µS/cm)	TDS† (ppm)	HCO_3^- (ppm)	Al (ppm)	Ca (ppm)	Cl (ppm)	F (ppm)	Fe (ppm)*	K (ppm)	Mg (ppm)	NO_3-N (ppm)	Na (ppm)	SO_4 (ppm)	Si (ppm)
Column 4 - Native Ground Water, Fast	3																	
	4		7.01	22.2	142.4	110.8	58	0.02	14	1.5	0.1	<0.005	5.1	3.6	1.7	5.4	7.1	14
	5		7.50	21.8														
	6		7.27	21.9														
	7		7.30	22.4														
	8		7.10	22.4														
	9	1.27E-04	7.25	22.7	138.9	109.5	57	0.02	14	1.9	<0.1	<0.005	4.7	3.4	1.6	4.8	7.3	14
	10	1.45E-04	7.26	22.2														
	11	1.22E-04	7.20	22.1														
	12	1.27E-04	7.13	22.0														
	13	1.74E-04	7.36	22.4														
	14	1.76E-04	7.71	22.6														
	15	1.51E-04	6.88	23.0														
	16	2.02E-04	7.03	22.4														
	17	2.17E-04	7.34	22.3	139.2	111.6	58	< 0.01	14	2.4	<0.1	<0.005	4.6	3.4	2.9	4.7	7.3	14
	Average		7.24	22.3	140.2	110.6	57	0.02	14	1.9	<0.1	<0.005	4.8	3.5	2.1	5.0	7.2	14
Column 5 - Native Ground Water, Med.	3																	
	4		7.15	22.4	151.8	111.5	57	0.02	14	2.6	<0.1	<0.005	5.1	3.5	2.2	5.3	8.0	14
	5		7.21	22.0														
	6		7.32	21.6														
	7		7.21	21.7														
	8		7.10	21.9														
	9	5.75E-05	7.01	22.0	138.5	109.6	56	0.01	15	1.9	<0.1	<0.005	4.7	3.5	2.4	4.8	7.2	14
	10	8.57E-05	6.95	21.5														
	11	7.69E-05	7.07	22.1														
	12	9.55E-05	7.04	21.8														
	13	9.01E-05	7.14	22.1														
	14	8.41E-05	7.20	21.7														
	15	8.45E-05	7.10	21.7														
	16	8.11E-05	7.09	21.4														
	17	9.15E-05	7.16	22.1	134.6	110.6	57	< 0.01	14	2.4	<0.1	<0.005	4.6	3.4	2.9	4.7	7.3	14
	Average		7.13	21.9	141.6	110.5	57	0.02	14	2.3	<0.1	<0.005	4.8	3.5	2.5	5.0	7.5	14

	Day	K (cm/sec)	pH	Temp (°C)	SpC† (µS/cm)	TDS† (ppm)	HCO_3^- (ppm)	Al (ppm)	Ca (ppm)	Cl (ppm)	F (ppm)	Fe (ppm)	K (ppm)	Mg (ppm)	NO_3-N (ppm)	Na (ppm)	SO_4 (ppm)	Si (ppm)
Column 6 - Native Ground Water, Slow	3																	
	4		7.32	22.2	142.3	115.0	63	0.01	14	1.7	0.1	<0.005	5.3	3.6	0.90	5.6	7.0	14
	5		7.32	22.0														
	6		7.32	21.9														
	7		7.04	22.5														
	8		7.28	21.9														
	9	1.87E-05	7.22	22.6	137.2	95.6	57	<0.01	14	2.3	<0.1	<0.005	4.9	3.5	1.2	5.2	7.3	<0.03
	10	1.72E-05	7.10	22.0														
	11	-	7.54	22.0														
	12	3.51E-05	7.30	21.7														
	13	3.51E-05	7.20	22.1														
	14	2.78E-05	7.32	21.8														
	15	7.32E-05	7.21	21.3														
	16	7.49E-05	7.62	21.7														
	17	6.16E-05	7.36	22.6	134.6	109.0	58	0.01	14	2.4	<0.1	<0.005	4.6	3.4	0.50	4.7	7.3	14
	Average		7.30	22.0	138.0	106.5	59	0.01	14	2.1	<0.1	<0.005	4.9	3.5	0.90	5.2	7.2	14
Belleville Aquifer Ground Water - Source	3		6.64	20.4														
	4		7.10	20.1	145.8	111.4	57	0.01	15	1.6	<0.1	<0.005	5.0	3.6	2.5	5.3	7.1	14
	5		7.07	20.0														
	6		6.92	20.3														
	7		6.81	20.1														
	8		6.93	20.8														
	9		6.76	21.2	138.4	111.9	58	0.02	15	1.5	<0.1	<0.005	4.6	3.5	3.3	4.8	7.2	14
	10		6.93	20.6														
	11		6.86	20.6														
	12		6.98	20.3														
	13		6.88	20.0														
	14		7.10	20.1														
	15		6.99	20.1														
	16		7.14	20.0														
	17		7.29	20.0	140.8	108.1	54	0.02	15	2.3	<0.1	<0.005	4.6	3.5	3.5	4.8	7.3	14
	Average		6.96	20.3	141.7	110.5	56	0.02	15	1.8	<0.1	<0.005	4.7	3.5	3.1	5.0	7.2	14

Table 5. Results of PHREEQC simulated mixing of ground water with surface water

Step	1	2	3	4	5	6	7	8	9	10	11
% Ground water	100	95	90	85	80	75	70	65	60	55	50
% Surface Water	0	5	10	15	20	25	30	35	40	45	50
Calcite SI	-1.40	-1.35	-1.30	-1.25	-1.20	-1.15	-1.11	-1.06	-1.02	-0.97	-0.92
Quartz SI	0.43	0.42	0.40	0.38	0.36	0.34	0.32	0.30	0.27	0.25	0.22
pH	7.10	7.12	7.14	7.16	7.19	7.21	7.24	7.26	7.29	7.31	7.34
Temp (°C)	20.1	20.1	20.1	20.1	20.1	20.1	20.1	20.1	20.1	20.1	20.1
$HCO3^-$ (ppm)	57	58	59	60	61	62	63	64	65	66	67
Al (ppm)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Ca (ppm)	15	16	17	18	19	21	22	23	24	25	26
Cl (ppm)	2	9	17	25	32	40	48	55	63	70	78
F (ppm)	0.1	0.1	0.2	0.2	0.2	0.3	0.3	0.4	0.4	0.5	0.5
K (ppm)	5.0	5.2	5.5	5.8	6.0	6.3	6.6	6.9	7.1	7.4	7.7
Mg (ppm)	3.6	3.6	3.7	3.8	3.9	4.0	4.1	4.2	4.2	4.3	4.4
NO_3-N (ppm)	2.5	2.4	2.3	2.2	2.1	2.0	1.9	1.8	1.7	1.5	1.4
Na (ppm)	5.3	10	14	19	23	28	32	37	41	46	50
SO_4 (ppm)	7.1	12	17	21	26	31	36	40	45	50	54
Si (ppm)	14	6.5	6.2	6.0	5.7	5.4	5.2	4.9	4.6	4.4	4.1
SAR	0.3	0.6	0.8	1.0	1.3	1.5	1.7	1.9	2.0	2.2	2.4
SP (%)	16.6	25.8	32.3	37.3	41.2	44.3	46.9	49.0	50.9	52.4	53.8

Step	11	12	13	14	15	16	17	18	19	20	21
% Ground water	50	45	40	35	30	25	20	15	10	5	0
% Surface Water	50	55	60	65	70	75	80	85	90	95	100
Calcite SI	-0.92	-0.88	-0.83	-0.78	-0.73	-0.68	-0.63	-0.57	-0.52	-0.46	-0.40
Quartz SI	0.22	0.19	0.16	0.12	0.09	0.05	0.00	-0.04	-0.10	-0.16	-0.24
pH	7.34	7.37	7.40	7.43	7.46	7.50	7.53	7.57	7.61	7.65	7.70
Temp (°C)	20.1	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0
HCO_3^- (ppm)	67	68	69	69	70	71	72	73	74	75	76
Al (ppm)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Ca (ppm)	26	28	29	30	31	32	33	35	36	37	38
Cl (ppm)	78	86	93	101	109	116	124	132	139	147	155
F (ppm)	0.5	0.5	0.6	0.6	0.7	0.7	0.7	0.8	0.8	0.9	0.9
K (ppm)	7.7	8.0	8.2	8.5	8.8	9.1	9.3	9.6	9.9	10	10
Mg (ppm)	4.4	4.5	4.6	4.7	4.8	4.8	4.9	5.0	5.1	5.2	5.3
NO_3-N (ppm)	1.4	1.3	1.2	1.1	1.0	0.92	0.82	0.71	0.61	0.50	0.40
Na (ppm)	50	55	59	64	68	73	77	82	86	91	95
SO_4 (ppm)	54	59	64	69	73	78	83	88	92	97	100
Si (ppm)	4.1	3.8	3.6	3.3	3.0	2.8	2.5	2.2	2.0	1.7	3.1
SAR	2.4	2.5	2.7	2.9	3.0	3.2	3.3	3.4	3.6	3.7	3.8
SP (%)	53.8	55.0	56.0	57.0	57.8	58.5	59.2	59.9	60.4	61.0	61.4

Appendix II. Hydraulic Conductivity Graphs

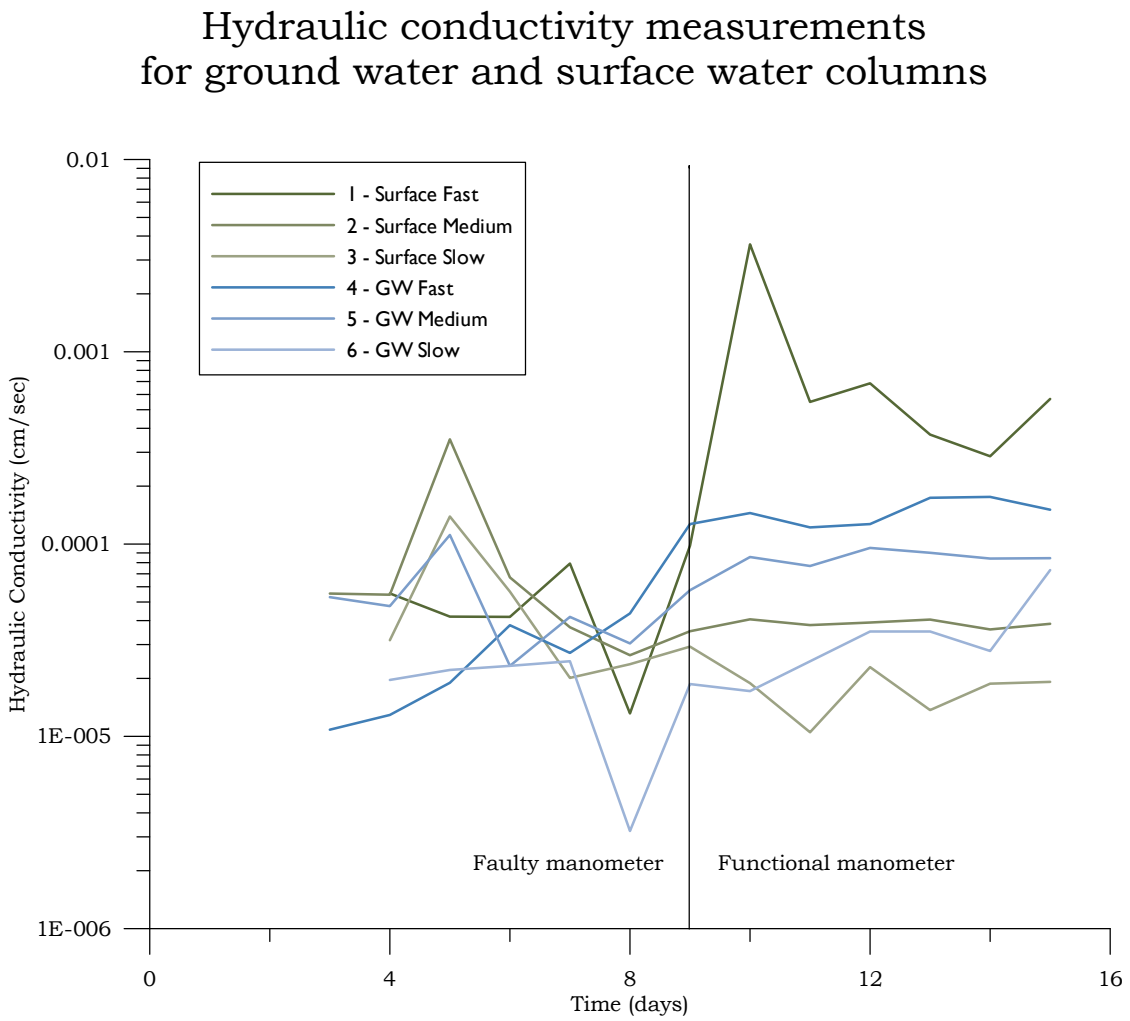

Figure 7. Hydraulic conductivity measurements for both ground water and surface water columns, day 3 through day 17

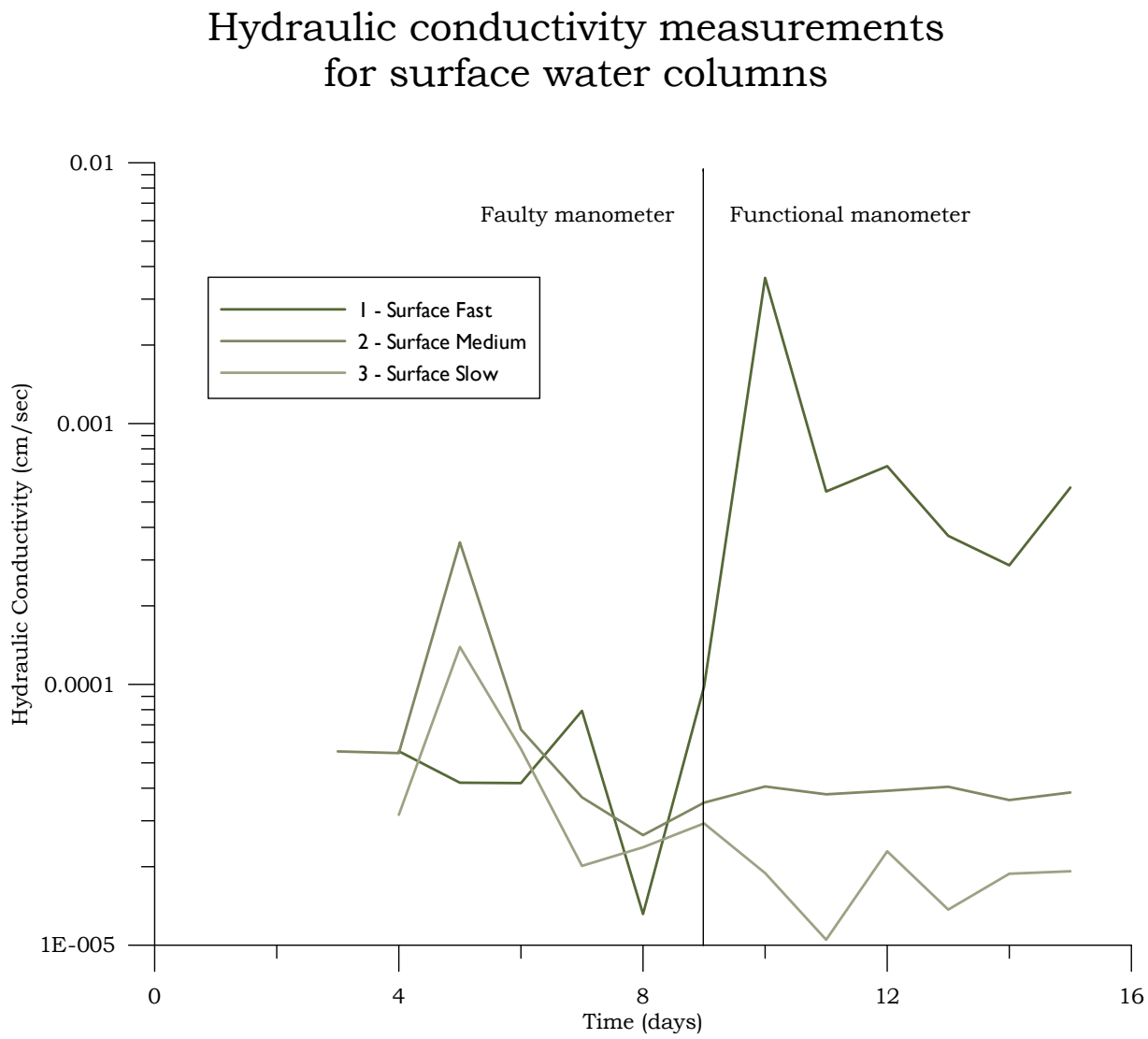

Figure 8. Hydraulic conductivity measurements for surface water columns, day 3 through day 17

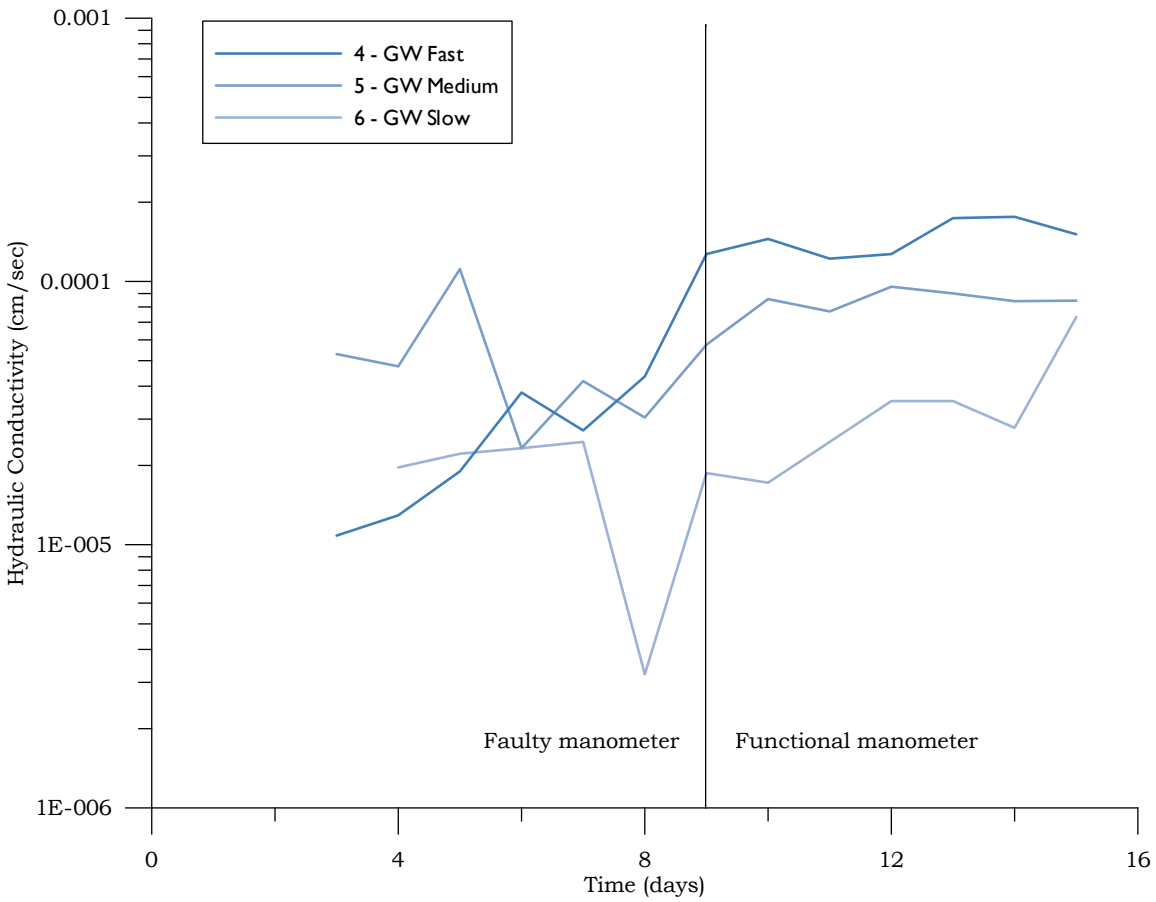

Figure 9. Hydraulic conductivity measurements for ground water columns, day 3 through day 17